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Costs and benefits of an enhanced reduction policy of particulate matter exhaust emissions from road traffic in Flanders

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Abstract

We demonstrate that accelerated policies beyond the steady improvement of technologies and the fleet turnover are not always justified by assumptions about health benefits. Between the years 2000 and 2010, particulate matter (PM) exhaust emissions from traffic in Flanders, a region of Belgium, will be reduced by about 44% without taking any extra reduction measures (baseline scenario). The PM emissions from road traffic were calculated using the MIMOSA model. Furthermore, we explored a range of options to increase attempts to reduce PM exhaust emission from traffic in 2010. When installing particle filters on heavy-duty trucks and buses, introducing biodiesel and diesel/hybrid cars, as well as slowing down the increase of private diesel cars, only an extra reduction of about 8% PM can be achieved in Flanders. The costs to achieve this small reduction are very high. To justify these costs, benefits for public health have been calculated and expressed in external costs. We demonstrate that only an enhanced effort to retrofit trucks and buses with particle filters has a net benefit. We have used Monte Carlo techniques to test the validity of this conclusion. It is concluded that a local or national policy that goes beyond European policies is not always beneficial and that additional measures should be assessed carefully.

Keywords: Particulate matter; Emission reduction strategies; Marginal costs; External costs; Air quality management

1. Introduction

Since the beginning of the 1990s, epidemiological studies have consistently shown that adverse health effects are associated with particulate matter (PM). Daily concentrations of PM_{10} and $PM_{2.5}$ are linked with cardio-respiratory health effects and even with mortality (Le Tertre et al., 2002; Atkinson et al., 2001; Katsouyanni et al., 2001). Cohort studies in the US have demonstrated that long-term exposure to ambient $PM_{2.5}$ is a particularly important factor in the reduction of life expectancy (Pope et al., 2002) and in lung cancer mortality. There are specific epidemiological studies on $PM_{0.1}$ that also find health effects (Wichmann and Peters, 2000). It is still difficult to show which component is responsible for the health effects and how, but a growing amount of evidence (WHO, 2005) points to traffic as the main source of adverse health effects due to air pollution. Moreover, there is growing concern that children's health is affected by traffic-related air

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pollution (Schwartz, 2004). It has also been established that lowering ambient concentrations or reducing certain contributions to ambient PM leads to lower health effects.

This evidence has motivated policy makers to tackle the problem of particulate air pollution. The Second Daughter Directive (1999/30/EC) on air quality established limit values for PM_{10} in the EU (European Commission, 1999). There are no emission limits for PM_{10} however, but in the Clean Air for Europe (CAFE) programme of the European Commission and in the framework of the UNECE Convention on Long-Range Transboundary Air Pollution (CLRTAP) possible emission targets are examined for primary particles. Meanwhile, the reduction of SO_2, NO_X and NH_3 agreed upon in the NEC Directive (2001/81/EC) will have a beneficial effect on PM concentrations (European Commission, 2001).

To meet European emission standards new vehicles are gradually becoming cleaner due to improvements in technology. The different technological generations (euro 0, euro 1, euro 2,…) result in a decrease in PM_{10} emissions from diesel vehicles.

We have explored a range of options to increase the effort of PM exhaust emission reduction from traffic in 2010 in Flanders, a region of Belgium. We have constructed a baseline scenario, taking into account current legislation up to 2010 as a baseline. We have derived costs for the different extra measures and calculated the benefits of reduced particulate air pollution in terms of reduced environmental external costs. For the latter we used the MIMOSA road traffic emission model (Mensink et al., 2000; Lewyckyj et al., 2004) to calculate the geographically distributed emissions for different measures, and applied the ExternE methodology (Friedrich and Bickel, 2001) to calculate the benefits. We then analysed which extra measures would be beneficial, i.e. where benefits exceeded the costs. We applied uncertainty analysis and Monte Carlo techniques to examine the robustness of the results, and we tested the sensitivity of the results to the inclusion of secondary effects of the measures on CO_2 emissions.

2. Methodology

We will first describe how we determined the baseline scenario for the year 2010. Different reduction measures that can be implemented to the Flemish vehicle fleet (until the year 2010) will then be described. We will proceed to indicate how we calculate the emissions of PM from road traffic: this will be done with the model MIMOSA. Finally, we will explain the methodology for evaluating the emission reduction for the different measures in terms of costs and benefits.

2.1. Description of baseline scenario

We used mobility data for the year 2000 and estimated traffic flows for the year 2010 of the Mobility Plan for the Flemish Region (MFR, 2001a), so that the baseline scenario would correspond to the Mobility Plans for the Flemish Region. No additional measures concerning modal shift and reduction of vehicle kilometres are assumed here (MFR, 2001b). We added recent initiatives like the implementation of particle filters on euro 2 local buses, the introduction of the euro 4 and euro 5 emissions standards and the introduction of low-sulphur fuels to this reference scenario. The vehicle fleet was updated with statistical data up to the year 2002. Together with updated preferences for diesel vehicles when buying a new car, this resulted in a share of about 74% of the total kilometres driven by passenger diesel cars in 2010 compared to 56% in 2002.

2.2. Description of emission reduction measures

Here we give an overview of the different emission reduction measures that we have evaluated (Schrooten et al., 2003). We also give the assumed implementation degree and the reduction percentage for the different emission reduction measures.

The first measure is the installation of particle filters in 25% of the euro 1, euro 2 and euro 3 heavy-duty trucks and buses by the year 2010. A particle filter reduces the PM emissions by 75–90% (Van Poppel and Lenaers, 2004; Durbin et al., 2003; van Ling et al., 2003). Depending on the remaining life of the vehicle at time of conversion, replacements of the filter element can be necessary. We assumed that a euro 1 and a euro 2 class vehicle requires one additional filter replacement, and that a euro 3 requires two replacements. The average cost (in EUR 2002 prices) of a particle filter that was taken into account is € 6250 (5500–7000) for euro 1, € 7500 (5500–9500) for euro 2 and € 11 500 (6000–11 500) for euro 3 heavy-duty trucks and buses. The range given starts from the minimum cost without replacement to the maximum cost with

two replacements. We took this variability into account in our assessment of costs and benefits.

The second measure considered is the introduction in 2010 of 5% biodiesel into the diesel mixture for the whole Flemish diesel fleet. As this mixture required neither new tanking facilities nor adaptation at vehicle level, it was considered to be a measure without implementation costs. A reduction of only 1% of $PM km^{-1}$ can be achieved (Verbeiren et al., 2003). The additional cost for biodiesel compared to diesel is due to increased production cost. This cost amounts to $0.00105\,€\,km^{-1}$ for private cars and vans, $0.00225\,€\,km^{-1}$ for heavy-duty trucks and $0.00750\,€\,km^{-1}$ for buses.

Over the last 10 years, there has been a large increase in private diesel vehicles in Flanders: from 35% in 1990 to 63% in 2002. Stopping this increase is the next measure that we evaluated. The Belgian Automobile Federation assumes that the engagements of the automobile industry to vehicles emitting lower CO_2 rates will result in a market for small diesel passenger cars (Peelman, 2003). The number of new private diesel cars in the year 2010 has therefore been held on the same level as in 2002, i.e. 63%—which corresponds to 70% diesel-driven passenger kilometres, as a further decrease does not seem realistic for Flanders. This stabilisation means that a fraction of car owners will have to be persuaded to buy a new gasoline car instead of a diesel car. The emissions of PM for a private gasoline car in the current fleet are small, compared to the (current) PM emissions for a private diesel car. We assessed that an average private gasoline car (in EUR 2002 prices) costs €390 less than an average private diesel car. The additional fuel costs (in EUR 2002 prices) of gasoline engines are €2.73 higher per 100 km than those of diesel engines, due to lower fuel efficiency and higher fuel taxes in Belgium.

The fourth measure is the introduction of diesel/hybrid cars. We assumed that by the year 2010, 25% of the new private diesel cars and new diesel vans will be diesel/hybrid vehicles. SAM and WRI (2003) state that a maximum of 15% of the new passenger cars could be hybrid vehicles in Europe by 2015, as a consequence of limitations in production capacity and market. Rault (2001) mentions that the market penetration of hybrid fuel cell, CNG and hydrogen for 2010, taken all together, will be less than 10%. Those penetrations do not take policy measures into account. We have assumed higher implementation levels because we have considered a subsidy for hybrid vehicles. However, this financial support has not been computed in the cost evaluation. Subsidies are transfers between economic agents and do not influence the cost for the community. A reduction of 50% in PM emissions per kilometre driven can be achieved (Verbeiren et al., 2003). The additional investment cost for a diesel/hybrid car with regard to a diesel car amounts to €485 (in EUR 2002 prices) and the lower fuel costs come to $7.20\,€\,100\,km^{-1}$ (in EUR 2002 prices).

Finally, a combination of all four previously additional measures was examined. Only for private cars, there is more than one measure that has an influence on the fleet composition, namely the third and fourth measure. We therefore recalculated the amount of diesel vehicles (measure 3) in the combined scenario. Based on this level, the penetration of diesel/hybrid cars is calculated (measure 4). The percentages of switch in vehicle fleet composition (measure 3 and 4), installation of particle filters (measure 1) and introduction of biodiesel (measure 2) are kept the same as in the individual measures.

2.3. The MIMOSA model

Based on the model derived by Mensink et al. (2000), the actual MIMOSA road traffic emission model has been extended and improved (Lewyckyj et al., 2004) and now calculates temporarily and geographically distributed traffic traffic emissions for all 16 major air pollutants from estimated traffic flows detailed by the number of vehicles and their speeds on each described road segment. The model uses the 'static' approach, i.e. hourly average speeds of the different vehicles. The emission factors used were partially extracted from experimental data collected by VITO ('on-road' measurements) as well as from the Copert-III report (Ntziachristos and Samaras, 2000). For missing data (some specific pollutants), emission functions from MEET (1999) were applied. In particular, PM emissions are calculated by using MEET (1999) emission factors. Distinctions are made between hot emissions, cold emissions and emissions due to evaporation (diurnal losses, running losses,...).

Vehicle categories are distinguished (Passenger Cars, Light-Duty Trucks (LDV), Heavy-Duty Trucks (HDV), Motorcycles and Mopeds) with further sub-categories depending on the age of the vehicle and its cylinder capacity for the passenger cars and for the two-wheeled vehicles, while age and weight criteria are used to distinguish LDV and

HDV (including buses and coaches). Distinctions are made between four fuel types (gasoline, diesel, LPG and 2-stroke gasoline), with lead and sulphur contents depending on the year of the simulation. Vehicle park compositions for the year 2010 were estimated by using the fleet module of the TEMAT model (Transport Emission Model to Analyse (non-) Technological measures). The TEMAT model forecasts vehicle fleet composition on the basis of mobility growth rates and survival fractions of the different vehicle categories (De Vlieger et al., 2001). Details on technologies and alternative fuel types are also given.

MIMOSA calculates geographically distributed hourly traffic emissions based on peak-hour (17–18) mobility input data from the official Flanders road traffic model. The MIMOSA model can work with any traffic model and is totally independent of the model used.

Finally, the time dependency of the emissions is simulated using normalised distribution factors expressing the fluctuations of the traffic flow as a function of the hour of the day, the day of the week and the month of the year. These distribution factors are directly derived from vehicle counting carried out in Flanders. Yearly traffic emissions are then calculated as the sum of all hourly traffic emissions.

2.4. Marginal cost curve

Reduction measures are best chosen as cost effective as possible, in other words as inexpensive as possible per unit of reduction. The marginal cost curve is a tool to make a cost effective selection between different reduction measures.

The marginal cost curve shows at what cost an additional emission reduction can be realised. Marginal cost curves plot the reduction potential on the X-axis against the cost per unit of reduction on the Y-axis for the different emission reduction measures. As the reduction percentage increases, the marginal costs increase because more and more expensive techniques have to be taken to obtain additional emission reduction.

The drawing up of the marginal cost curve supposes an iterative calculation procedure, because some measures interact with each other. The starting point is the reduction measure with the lowest unit cost: this is the first point of the marginal cost curve. Next, all remaining measures are compared according to the ratio of additional yearly cost to additional reduction potential. The cheapest in this series of alternative reduction measures becomes the second point on the curve, and so on. With the determination of the extra reduction potential, the reduction already realised through previous measures is taken into account: the reduction percentage has to be taken onto the remained emission level.

The cumulative cost is presented on the Y-axis. The cumulative cost is the sum of the total costs of all reduction measures that are taken into account in the cost curve. The total cost of a reduction measure is the product of the unit cost and the amount of emission reduction.

The marginal cost curve therefore indicates which reduction measures have to be taken first to realise an additional emission reduction in a cost-effective way.

2.5. The ExternE methodology

The European ExternE project provides an accounting framework based on an impact pathway methodology. It basically follows a pollutant from its emission until it causes an impact or damage. It allows monetisation of this environmental damage from air pollution for a specific technology and trajectory. More detailed descriptions of this methodology can be found in Friedrich and Bickel (2001). This bottom-up approach results in a marginal external cost of a transportation technology. Marginal externalities from fossil-fuelled transportation technologies are dominated by the public health impacts from emissions of primary $PM_{2.5}$, which are especially important for diesel-fuelled vehicles, in urban traffic and from nitrate aerosols (following emissions from NO_X). The uncertainty of the external costs is documented in Rabl and Spadaro (2001) and studied in detail for Belgium in Int Panis et al. (2002), but is not discussed further here. We have included Monte Carlo techniques (performed with the commercially available Crystal Ball© software of Decisioneering on a desktop computer) to study the parametric uncertainty of aggregating the marginal external costs of all motorised road transportation modes to the national level (Int Panis et al., 2004). We have made a distinction between a more rural situation and an urban one, with higher external costs per tonne of $PM_{2.5}$ in urban areas: 430 € $tonne^{-1}$ (95% confidence interval 210–650 € $tonne^{-1}$). The rural area includes the most highways, due of the fact that

the population density in these two situations is comparable. Here external costs are 106 € $tonne^{-1}$ (95% CI: 52–160 € $tonne^{-1}$) These external costs per tonne of $PM_{2.5}$ have been used to analyse the benefits of different measures compared to the baseline scenario.

3. Results

3.1. Emissions

Exhaust PM emissions from road traffic represent 37% of the total amount of PM emissions with an aerodynamic diameter of less than 2.5 μm ($PM_{2.5}$) in Flanders in the year 2000.

Due to the disappearance of the oldest and at the same time most polluting diesel vehicles, there will be a reduction of 44% in PM towards the year 2010 in the baseline scenario. The additional reduction that can be achieved by the introduction of particle filters, biodiesel, diesel/hybrid cars and slowing down the increase of private diesel cars is small (−8%).

3.2. Marginal costs and benefits

In Table 1, the results of the iterative process to work out the cost curve, the marginal costs (at a discount rate of 5%) and the reduction potential for the different emission reduction measures, are presented.

The marginal costs for the different measures are very high. Slowing down the amount of newly bought euro 3 private diesel cars in favour of gasoline cars would have been the most cost-efficient measure at the time the study was done in 2002–2003. The measure was not implemented, however, due to lack of time (since 2005, the euro 4 standard has been implemented) and due to doubts over the expected increase in CO_2 emissions.

It is more efficient to equip heavy-duty trucks with a particulate filter, starting with euro 1 heavy-duty trucks, before placing particle filters on newer heavy-duty trucks or buses. Retrofitting older vehicles is more cost effective than newer vehicles because of the higher reduction potential for older vehicles, while the annual costs remain

Table 1
Marginal costs and benefits for the different reduction measures, ranked according to marginal cost and taking into account the dependency of measures

Measure	Vehicle category	Euro-standard	Reduction potential (tonne)	Marginal cost (€ kg^{-1}) (95% CI)	Marginal benefit (€ kg^{-1}) (95% CI)	Cost-benefit ratio (95% CI)	Probability below 1 (%)
Replace diesel by gasoline	Passenger cars	Euro 3	46	46.5	205 (125–283)	0.23 (0.16–0.36)	100
+ Install particle filter	Heavy duty trucks	Euro 1	+11	74.5 (65–82)	140 (83–193)	0.53 (0.36–0.88)	100
+ Install particle filter	Heavy duty trucks	Euro 2	+25	125 (93–156)	140 (83–193)	0.88 (0.56–1.6)	68
+ Install particle filter	Heavy duty trucks	Euro 3	+63	161 (88–234)	140 (83–193)	1.14 (0.57–2.23)	36
+ Install particle filter	Local buses	Euro 3	+1	270 (147–393)	406 (200–611)	0.66 (0.3–1.5)	85
+ Install particle filter	Travel buses	Euro 2	+0,3	513 (383–643)	187 (118–256)	2.8 (1.7–4.7)	0
+ Install particle filter	Travel buses	Euro 3	+1	602 (328–875)	187 (118–256)	3.2 (1.6–6.1)	0
+ Replace diesel by hybrids	Passenger cars	Euro 4	+55	644	205 (125–283)	3.10 (2.3–5.0)	0
+ Replace diesel by gasoline	Passenger cars	Euro 4	+75	690	205 (125–283)	3.40 (2.4–5.4)	0
+ Replace diesel by hybrids	Vans	Euro 4	+6	721	203 (120–293)	3.60 (2.4–5.9)	0
+ Introduce biodiesel	All categories	All	+36	1 936	192 (118–264)	10.1 (7.3–15.9)	0
Total			320				

The 95% uncertainty interval is calculated with Monte Carlo methods.

approximately the same for new and old vehicles. For a similar reason, retrofitting buses is less efficient than retrofitting heavy-duty trucks. Buses travel fewer kilometres per year than heavy vehicle trucks; their reduction potential is therefore less, while the annual costs stay approximately the same. This is even more pronounced for travel buses because they drive for even fewer kilometres per year than local buses.

The replacement of new euro 4 private diesel cars with private gasoline cars or private diesel/hybrid cars would be a very expensive option, as would the replacement of new diesel vans with diesel/hybrid vans and the introduction of biodiesel.

Overall, the total PM emission reduction with these additional measures is 320 tonne (−8%). This is small compared to the predicted reduction of about 44% under current legislation from 2000 to 2010.

The total costs for the combination of all measures to achieve this are 178 billion € $year^{-1}$.

For the same combination of measures, the external costs have been computed. Here we used the marginal external costs per tonne of $PM_{2.5}$ emitted from Table 1, and the geographical distribution of the different vehicle categories considered. With local buses primarily in urban areas and heavy-duty trucks for the greater part on highways for example, this generates a location-weighted marginal external cost per vehicle category.

Thus, the marginal benefits are the same on a per tonne basis for the same vehicle categories, irrespective of their euro class, because similar vehicle categories have the same geographical distribution of mileage.

The cost–benefit ratio for replacing new diesel by new gasoline euro 3 cars is the lowest. This measure and retrofitting euro 1 heavy-duty trucks with particle filters are both cost beneficial in all of the circumstances considered. This means that even when the uncertainty about the external costs per tonne of $PM_{2.5}$, and about the number of times the particle filters have to be replaced, there is 100% certainty that the cost–benefit ratio will be below one. When retrofitting euro 2 heavy-duty trucks with particle filters, there is a 68% probability of having a cost–benefit ratio of below one (Fig. 1). However, retrofitting euro 3 heavy-duty trucks is only beneficial with a probability of 36%, meaning that it is more likely that costs will be higher than benefits under the given assumptions. Due to the fact that local buses (euro 3) have a marginal external cost that is almost three times higher than heavy-duty trucks per tonne of $PM_{2.5}$, and only an increased cost per tonne of about 70%, this reduction measure would again be more likely to be beneficial. All other measures in Table 1 have very high costs and their cost–benefit ratio is significantly higher than 1.

4. Discussion and conclusions

This paper describes a case study where a predefined set of measures to reduce PM emissions from traffic has been analysed with respect to their costs and benefits. The study was intended to inform policy makers how great the extra reduction of PM emissions in 2010 could be compared to the reference scenario, and whether there was a net benefit to society.

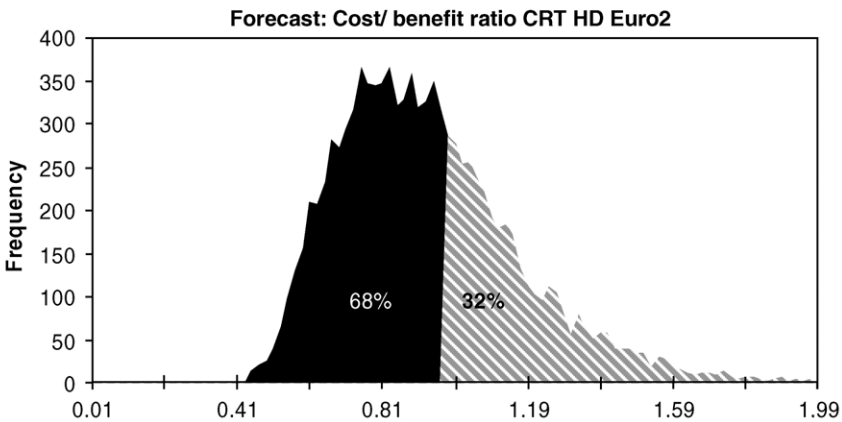

Fig. 1. Results of the Monte Carlo analysis.

4.1. Which individual measures are beneficial?

Individual measures are beneficial for particle filters and for the euro 3 gasoline initiative. Hybrids and biodiesel impose more costs than benefits. Biodiesel is not a PM policy measure as such, but more a measure against global warming. The cost of hybrids is an obstacle to this measure. More financial efforts are necessary to introduce hybrid diesels into the market. The Belgian authorities recently introduced a tax exemption for new hybrid vehicles as a policy measure to tackle climate change. So far, no hybrid diesels have been introduced.

Encouraging people to buy gasoline cars (for instance by stimulating financially the price of gasoline) would have been a very good measure for euro 3 cars, but there was too little time to implement this, as the euro 4 standard was in place by 2005.

4.2. Does the analysis change when other pollutants are included?

The costs of the different measures were allocated to PM, although for some measures this is arguable. The introduction of biodiesel and hybrid vehicles is driven by concerns about greenhouse gas emissions. This might be a reason to introduce the measure, because of its perceived benefit to tackle climate change. On the other hand, replacing diesel vehicles by gasoline in general increases CO_2 emissions, because diesel engines have a lower consumption rate of fuel per kilometre driven than gasoline engines. In consequence, replacing new euro 3 diesel cars with gasoline cars could have resulted in a net cost to society instead of the calculated benefit. In order to test these sensitivities, we checked the effect of the measures on CO_2 in a quantitative manner. For example, when retrofitting euro 1 trucks with particle filters, CO_2 emissions increase up to 5% for these trucks due to a reduced fuel efficiency (Van Poppel and Lenaers, 2004). For euro 1 trucks, the number of trucks that will be retrofitted in 2010 is 780. Their yearly CO_2 emission averaged over all weight categories, mileage and driving conditions of heavy-duty trucks is 20.3 tonnes. The expected PM reduction was about 11 tonne. If we assume an external cost of 20 € $tonne^{-1}$ of CO_2, that has also been used by ExternE as an upper limit for the marginal external costs, the expected secondary damage or benefit from a 5% CO_2 increase, when retrofitting heavy-duty trucks would be approximately 1.8 € $tonne^{-1}$ of PM reduction. When we take this into account in the cost–benefit ratio in Table 1, the result does not change and the conclusion that retrofitting euro 1 trucks is beneficial, is robust against assumptions about secondary CO_2 effects. Moreover the assumption of an external damage cost of 20 € $tonne^{-1}$ is not critical. Even with an extreme assumption that the damage costs for CO_2 are in the order of 200 € $tonne^{-1}$ CO_2, the secondary damage would be about 5 € $tonne^{-1}$ PM reduction, and the probability of the cost–benefit ratio being lower than 1 would still be 97%.

We have done the same for the installation of particle filters in euro 2 and euro 3 heavy-duty vehicles. This resulted in a probability of 63% and 30% for euro 2, respectively, euro 3 of being cost beneficial. This decrease in probability due to secondary effects is not likely to influence the overall conclusion on the benefits of retrofitting heavy-duty trucks with particle filters. With respect to buses, the results in Table 1 again do not alter when CO_2 effects are added. The positive CO_2 shift when introducing hybrids instead of diesel is too small to have an impact on the cost–benefit ratio.

The average CO_2 reduction for the replacement of a diesel car by a diesel/hybrid car is approximately 0.93 tonnes. The expected PM reduction was about 0.62 kg $vehicle^{-1}$. If we assume an external cost of 20 € $tonne^{-1}$ of CO_2, the expected secondary damage or benefit from a 20% CO_2 decrease, when retrofitting replacing a diesel car by a diesel/hybrid car would be approximately 30 € kg^{-1} of PM reduction. The PM and CO_2 benefits together are 253 € kg^{-1} of PM reduction. The cost/benefit ratio in Table 1 does not change significantly.

Finally, at both ends of the cost–benefit spectrum, the measures are insensitive to the inclusion of CO_2 effects. The increase in fuel consumption when replacing diesel cars with gasoline cars, nor the introduction of almost CO_2 neutral biodiesel is going to influence the ratios in Table 1.

4.3. It is a comparison of annual costs and annual benefits

Both costs and benefits have been expressed on an annual basis. For costs, this means a total discounted cost divided by the lifetime expectancy of the vehicle. For benefits, only impacts in that year are considered. For acute health effects, this is straightforward; for health effects from chronic

exposure, we used the ExternE approach where the reduction of particulate air pollution in a given year is transformed into a number of life-years saved in a population (Friedrich and Bickel, 2001).

4.4. Do other studies confirm this?

Mediavilla-Sahagún and ApSimon (2003) calculated the annualised benefit of retrofitting all buses in the London area with particle filters. For every million pounds spent on the measure, 23–94 premature deaths were reduced. Their health benefit calculation is based on the same epidemiological studies as in ExternE, and the risk of premature death increase used is adjusted in much the same way as was done in Friedrich and Bickel (2001), on which we base our benefit estimates. In the ExternE methodology, however, the number of life years rather than the number of deaths are calculated and valued. Here we used the central estimate for chronic impacts from Friedrich and Bickel (2001) of about 100 000 € per life–year lost. If we assume a life reduction of 5–10 years per case due to premature mortality caused by particulate air pollution, and an exchange rate of 1.4 £ $€^{-1}$, this generates an estimated cost–benefit ratio of 0.02–0.12. Considering the fact that secondary aerosols were quantitatively included in the study of Mediavilla-Sahagún and ApSimon (2003), and that only 25% of trucks and buses would be fitted with particle filters in Flanders, this is a good agreement with the cost–benefit ratios in Table 1.

4.5. Policy implications

Without an emission ceiling for primary particles, there is no real political incentive to reduce primary particulate emissions. Enhanced policies, in order to accelerate the reduction of emissions above the baseline improvement are costly and do not always result in an overall benefit. The steady improvement in technology and the slow but steady turnover of vehicle fleet has a much greater impact on particle emissions. A policy aimed at stimulating the switch to gasoline cars would have been effective for euro 3 cars. However, this policy study also concludes that the time for its implementation was too short. Retrofitting particle filters is nevertheless a good measure to implement. This policy study did not consider other pollutants quantitatively. It is necessary to derive overall cost–benefit estimates for several pollutants at the same time, to deal with trade-offs like the one between short-term effects of particulate air pollution and long-term global warming issues. More effort should be done to make integrated assessment studies of air quality policies, taking all important pollutants into account.

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